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COLD REGIONS RESEARCH AND ENGINEERING LABORATORY
HANOVER, NEW HAMPSHIRE

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Treatment of primary sewage effluent by rapid infiltration

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M.B. Satterwhite, B.J. Condike and G.L. Stewart

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Treatment of unchlorinated primary sewage effluent by using rapid infiltration basins resulted in a high degree of wastewater renovation in a humid, cool northern climate. Inundating 9 treatment basins for 7 days followed by 14 days of rest, from 4 January to 21 June 1974, resulted in effluent additions totaling about 27 m. Analysis of the groundwater from the treatment site and from the peripheral area showed that total coliform bacteria, 5-day biochemical oxygen demand, and chemical oxygen demand were essentially removed, while phosphorus concentrations were only one-third of the applied effluent concentrations. Total nitrogen additions to the treatment basins during the 7-day inundation period were about 54% greater than the nitrogen additions in the 1973 investigations. Even so,

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groundwater nitrogen concentrations were closely comparable to those observed in the 1973 study. Efforts to increase nitrogen removal through longer inundation periods resulted in a gradual decrease in the infiltration capacities of the basins. Calculation of the organic matter additions strongly suggested that the reduced infiltration rates resulted from surface clogging. This study showed that proper management is needed if rapid infiltration basins are used for nitrogen removal by maintaining effluent infiltration in northern climates.

PREFACE

This report was prepared by Melvin B. Satterwhite, Agronomist, New England Division, Corps of Engineers; Brian J. Condike, Chief Chemist, Water Quality Laboratory, New England Division, Corps of Engineers; and Gordon L. Stewart, Soil Scientist, Department of Plant and Soil Science, University of Massachusetts, Amherst, Mass.

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SUMMARY

Inundation of rapid infiltration basins for 7 days followed by recovery of 14 days continued to renovate unchlorinated primary sewage effluent to a better level than that achieved by conventional secondary treatment; the level was closely comparable to the level of renovation achieved by tertiary treatment. Analysis of groundwater from the application site and the peripheral area showed that total coliform bacteria, 5-day biochemical oxygen demand, and chemical oxygen demand were essentially removed. Phosphorus levels were about one-third of the effluent concentration; however, they were substantially less than the levels in the effluent from conventional secondary treatment systems.

The total nitrogen levels of groundwater were not reduced to less than 10 mg/l in the wells which were strongly influenced by the percolate from the treatment site; these wells continued to have nitrogen levels from 10 to 20 mg/l. These levels give a different perspective when total nitrogen additions to the treatment basin are considered. During 1974, nitrogen additions were about 55 kg/ha day, which was 54% greater than the 36 kg/ha day total nitrogen applied when the basins were inundated for only 2 days. Although nitrogen additions were larger during the 1974 study, a proportional increase in groundwater nitrogen levels was not observed.

The actual nitrogen removal mechanism was not determined; but the substantial losses observed were believed to be the consequence of denitrification and volatilization. This seems reasonable in view of the large nitrogen inputs and small percentages accountable in plant uptake, soil fixation or accumulation of resistant nitrogenous organics.

Efforts to increase nitrogen removal by lengthening the inundation period resulted in a gradual decline in the basin infiltration capacity over several months. Calculations of the organic matter additions strongly indicated that the reduced infiltration rate resulted from clogging of the surfaces of the basins by accumulating organic matter. The anomaly is that both long inundation periods and available organic matter are necessary for denitrification. These same factors were mutually operative in reducing the infiltration rate. The relationships of these factors, which ensure the rejuvenation and maintenance of infiltration capacity in humid, cool climates and reduce nitrogen, require additional evaluation.

The results of this study provide guidance for determining the optimal amount of effluent that may be applied in a land treatment system. The results of such determination, in turn, may provide 1) a reduction in land area required for treatment, resulting in lower costs; and 2) a means for military facilities commands to better assess the possible economics of land treatment as an alternative to more conventional wastewater treatment methods. The results may also provide a reduction in capital, operations and maintenance costs.

TREATMENT OF PRIMARY SEWAGE EFFLUENT BY RAPID INFILTRATION

M.B. Satterwhite, B.J. Condike and G.L. Stewart

INTRODUCTION

Treatment of unchlorinated primary sewage effluent by rapid infiltration or high-rate groundwater recharge has been practiced at Fort Devens, Massachusetts, for over 30 years. Previous investigation of this treatment facility revealed that unchlorinated primary effluent was being renovated to a quality better than that generally achieved by conventional secondary wastewater treatment. However, although effluent was substantially renovated, nitrogen levels in groundwater beneath the treatment site and the surrounding area often exceeded nitrate levels permitted for drinking water.[20]

Nitrogen (as NO_3-N) movement from rapid infiltration systems has been reported for other rapid infiltration sites.[3 4 6 9 15] When groundwater recharge and additional treatment of wastewater effluents have been major considerations, efforts have been made to reduce nitrogen levels in the applied effluent. Manipulation of application rates, application periods, and levels of readily available organic carbon have been explored with varying success. Emphasis has been given to the reduction of nitrogen by enhancing the nitrification-denitrification processes. Conditions needed to attain the desired nitrogen transformation have been discussed at some length by various authors.[10 12 13 20 21]

Nitrogen reduction in rapid infiltration systems requires simultaneously the nitrification of ammonium nitrogen (NH_4-N) and the denitrification of nitrate-nitrogen (NO_3-N). Some nitrogen may be taken up by microorganisms and surface vegetation or through soil fixation. The amount of nitrogen removed by these processes generally constitutes only a very small percentage of the total nitrogen additions to high-rate loading systems.

Previous studies at the Ft. Devens treatment facility have shown that most of the groundwater nitrogen was composed of nitrate-nitrogen.[20] Comparison of nitrogen levels in the effluent and those in the groundwater showed that substantial nitrogen reductions had occurred which were assumed to be the result of denitrification. Even so, the nitrogen levels averaged 10 to 20 mg/l in the groundwater surrounding the rapid infiltration site.

Laboratory and field studies in arid climates have shown that applications of secondary effluent for periods of 2 to 5 days were conducive to nitrification but were limited in their capacity to reduce the percentage of total nitrogen from the secondary effluent.[3 6]

Other studies have shown that longer inundation periods have increased nitrogen removal, with further increases by additions of readily available organic matter, e.g., methanol or glucose, to the effluent.[9 11 13 19] These studies have shown that greater nitrogen removal in rapid infiltration basins was possible through practices that enhanced the nitrification-denitrification processes. However, because nitrogen removal by denitrification is a biological process dependent upon many environmental factors, it is nearly impossible to extrapolate operation experience and data from one climate to another.

The objectives of the 1974 studies at the Fort Devens sewage treatment facility were to remove greater amounts of nitrogen than had previously been removed from the primary sewage effluent by management of the treatment system and to monitor groundwater quality. Greater amounts of nitrogen were removed by changes in the operation and management of the treatment basins, including the use of longer inundation periods.

MATERIALS AND METHODS

Unchlorinated primary sewage effluent was applied to the treatment basins from 4 January 1974 through 21 June 1974 by using an effluent application cycle of 7 days inundation followed by a recovery period of 14 days.

During January and February, 12 treatment basins were used, although a total of 22 basins were available. Effluent during this period was applied to three sets of four treatment basins,* each set totaling about 1.2 hectares (3.0 acres) (Fig. 1). Rates of daily effluent application to the basins were 0.35 to 0.42 m/day (1.1 to 1.4 ft/day). After application of effluent to the 4 basins for 7 days, effluent accumulations on the basins were generally 50 to 60 cm in depth, but infiltrated the gravelly, medium sand surface horizons of the treatment basins within the first three or four days of the recovery period.

Observations during this two-month period showed that the time required for the applied effluent to move into the basins was not substantially different from that of the previous year's operation. Because of the foregoing findings and the improvement in groundwater quality surrounding the application area, the daily application rate was then increased. Since effluent flows to the basins were not controllable, the application rate was increased by using 9 instead of 12 treatment basins, or 3 treatment basins per set.† These treatment basins were used throughout the study. Daily rates of application to the 9 basins were 0.42 to 0.57 m/day (1.4 to 1.9 ft/day).

Effluent flows during the 1974 study were 3070 to 6060 m^3/day (0.81 to 1.6 million gallons per day) with an average flow of 4470 m^3/day (1.18 mgd) (Fig. 2). The total effluent applied from 4 January through 21 June 1974 was about 27 m.

Groundwater quality beneath the treatment basins and the surrounding area was monitored by collecting and analyzing liquid samples from 20 groundwater observation wells. All wells were installed by wash drilling, using either skid-mounted or truck-mounted diamond core drills. All holes were cased during drilling operations. Tapwater was used in the installation of the well casings to remove substrate from the casings during drilling to prevent possible contamination.

** Refs. 5, 6, 7, 8, 9, 10, 11, 13, 14, 15, 16, 17.*
† Refs. 7, 8, 9, 10, 11, 13, 14, 15, 16.

Following installation, each well was pumped for 30 minutes to remove fine sediments, and then covered by a protective cap. Each well screen was positioned so that the upper portion of the water table could be sampled throughout the hydrologic year. Dimensions of the 20 wells and an illustration of a typical well are presented in Figure 3.

Groundwater samples were collected biweekly from each observation well. The large wells, which had deep water tables, were sampled with a 0.4-l (0.10-gal) Kemmer water sampler. Samples from the 3.2-cm-diam wells, which had shallow water tables, were collected directly into sterile 1-l (0.26-gal) labeled fleakers using a hand-operated suction pump.

A composite (18-l) sample of the primary sewage effluent was collected during the 24-hour period preceding each biweekly sampling, using a proportional sampler equipped with a refrigeration unit that cooled the composite sample to 4°C. Following thorough mixing of the 18 l of composited effluent, a liter subsample was drawn off for laboratory analysis.

Groundwater and composite effluent samples were transported to the laboratory within four hours after collection, and subsequently analyzed for the following parameters according to procedures outlined in *Standard Methods for Examination of Water and Wastewaters*[2]: pH (glass electrode method); 5-day biochemical oxygen demand (BOD_5) (Winkler determination); chemical oxygen demand (COD); organic Kjeldahl nitrogen (potentiometric titration); ammonia nitrogen (Nesslerization); total and ortho-phosphate (PO_4-P) (stannous chloride method); and total coliform and fecal coliform bacteria (membrane filter technique). Nitrate nitrogen (NO_3-N) and chlorides were determined utilizing specific ion electrodes.[14,17]

To monitor percolate movement at the treatment site, tritium tracer investigations were conducted during late March and early April 1974. The primary sewage effluent applied to treatment basins 10, 15 and 16 was spiked with tritium tracer to a concentration of 3×10^3 μc/ml or less, which was about 2000 counts/min above background levels on the liquid scintillation counter. A total of 75 curies of tritium was applied over a 4-day spiking period.

Groundwater samples for tritium analysis were obtained at frequent intervals during the test period, initially three times a day, and then once a day. Samples were collected from 18 observation wells around the perimeter of the treatment site, from the nearby ponds,

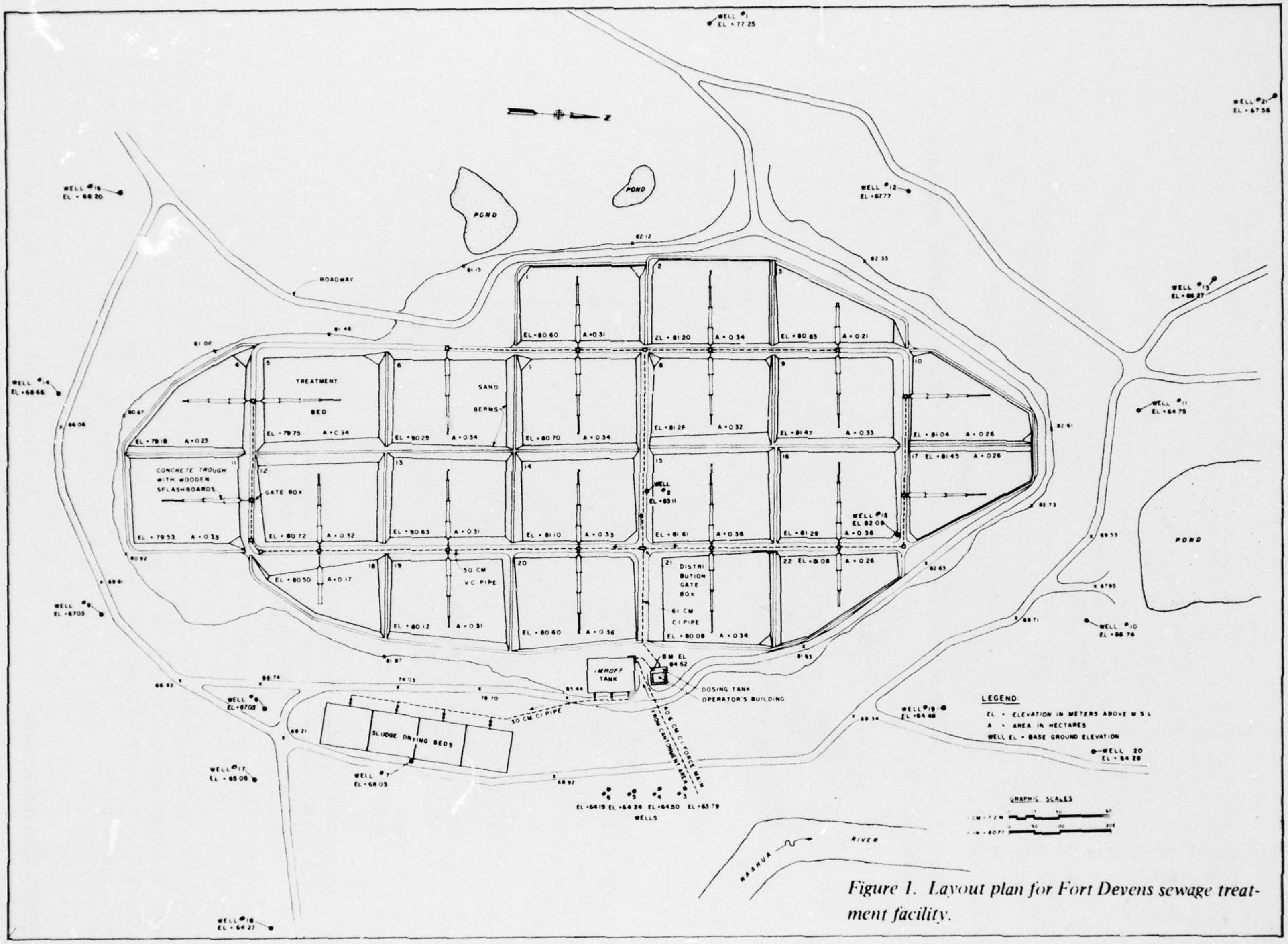

Figure 1. Layout plan for Fort Devens sewage treatment facility.

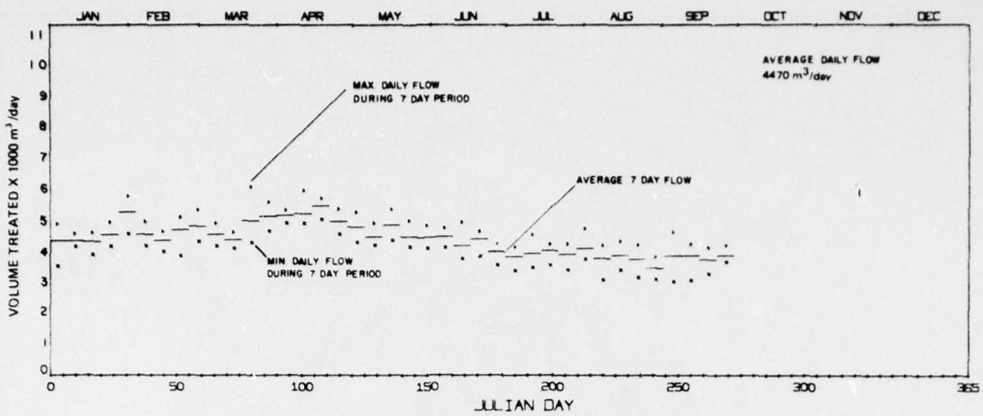

Figure 2. Maximum, minimum and average 7-day wastewater flows to Fort Devens sewage treatment facility in 1974.

Well	Total well length (m) A	Well screen length (m) B	Riser pipe above ground (m) C	Well diameter (cm) D
1*	13.0	1.5	0.6	7.6
2*	20.5	1.5	1.0	15.2
3*	3.8	0.9	0.9	3.2
4*	5.9	0.9	0.9	3.2
5*	7.3	0.9	0.9	3.2
6*	8.8	0.9	0.9	3.2
7	9.6	3.1	0.9	3.2
8*	6.7	0.9	0.9	3.2
9	7.0	3.1	0.8	3.2
10	7.9	3.1	1.0	3.2
11	5.5	3.1	0.9	3.2
12	8.8	3.1	1.0	3.2
13	9.1	3.1	0.9	3.2
14	8.5	3.1	1.0	3.2
15		To be installed		
16	6.2	3.1	0.9	3.2
17	6.4	3.1	0.9	3.2
18	6.7	3.1	1.0	3.2
19	8.0	3.1	1.2	3.2
20	6.0	3.1	0.9	3.2
21	8.3	3.1	1.0	3.2

* Metal well screens and riser pipes; all other wells consisted of polyvinyl chloride well screens and riser pipe.

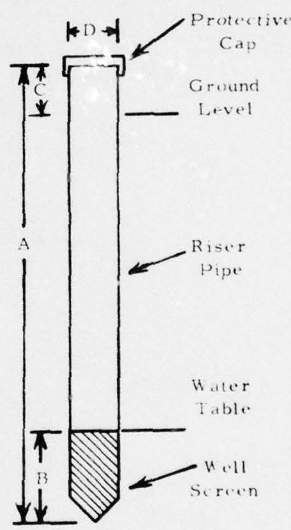

Figure 3. Diagram and dimensions of 20 groundwater observation wells.

and from the spiked effluent entering the treatment basins. All samples were analyzed in the laboratory using standard liquid scintillation counting techniques to measure tritium activity.

RESULTS AND DISCUSSION

Field observations at the conclusion of these investigations showed that the rejuvenation of the basins' infiltration capacities was not achieved during the 14-day recovery period. Previous field studies had shown that the initial infiltration rates of the basins were about 71 cm/hr at the beginning of the application period, but decreased rapidly to about 2 cm/hr after 6 hours of effluent application.[20] The consequence of the decreased infiltration rates was the accumulation of effluent on the basins' surface to a depth of 50 to 70 cm. Although the infiltration rates were quite low, the ponding created a positive pressure, which increased the infiltration rates.

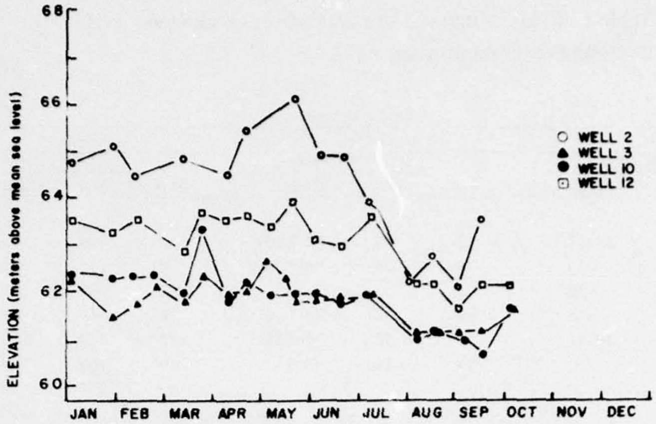

Figure 4. Water table elevations in selected wells during 1974.

Initially, the effluent accumulated on the treatment basins infiltrated the basins within the first 4 to 5 days following the application period. After four months operation, this time had increased to about 7 to 9 days. By the end of this study, the effluent applied to several basins failed to infiltrate completely the basins' surfaces before the beginning of the next inundation cycle.

The daily additions of organic material as BOD_5 were calculated to be about 147 kg/ha day. This was comparable to the levels reported in the literature at which organic matter causes the failure of sand filters when sufficient recovery periods are not provided.*

After the 9 treatment basins were rested for 60 consecutive days, primary effluent was again applied using the same 7-day application/14-day recovery operational cycle. Observations during the first full operation cycle showed that the basin infiltration capacities had been rejuvenated during the extended recovery period. These observations tend to support the conclusion that the materials responsible for reduced infiltration were organic.

The sulfide precipitates which have reduced the infiltration rate in other infiltration studies were not found in the treatment basins used here.[11]

Although quantitative data describing the impacts of the present operation cycle upon the basins' infiltration rates were not collected, observations made during this study provided empirical evidence that the 7-day inundation/14-day recovery operation cycle followed was not ideal for long-term operation of these treatment basins. This conclusion is supported by previous works which showed that continuous inundation of rapid infiltration basins gradually increased the infiltration capacity to negligible levels. When intermittent recovery periods of adequate duration were provided, clogging materials such as organic matter and sulfide precipitates were oxidized and the infiltration capacity was partially or completely restored.[5 7 16]

Groundwater movement at the treatment site was in a north and northeast direction, as determined from groundwater gradients and water quality in the various observation wells. Water table elevations were variable in all wells throughout the study. Maximum water table elevations were observed early in June and minimum elevations were observed in late September. Data for well 2 showed a mounded water table beneath the treatment site, which was 0.9 to 3.7 m higher than water tables for wells 3, 10 and 12, in the surrounding area (Fig. 4). Water table elevations shown for wells 3, 10 and 12 were representative of those observed for other wells at the treatment site.

Groundwater gradients from well 2, beneath the treatment site, to the other wells varied as water tables fluctuated throughout the study period, but they indicated that groundwater movement was in the north and east directions towards the Nashua River. Groundwater gradients in wells 11, 12, 13 and 21, west of the treatment site, showed groundwater movement from the general direction of wells 12 and 21 towards well 11, and in turn from well 11 towards the nearby lake and the Nashua River (Fig. 1).

Chloride levels in the groundwater were used to monitor groundwater movement because the negatively charged chlorides are not readily fixed or

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* *Refs. 1, 5, 7 and 22.*

Table I. Relative impacts of percolate upon chloride levels in groundwater monitoring wells.*

	Relative impact †							
	Slight			Moderate			Great	
	Level (mg/l)			Level (mg/l)			Level (mg/l)	
Well	Mean	Range	Well	Mean	Range	Well	Mean	Range
1	31	24-52	3	92	42-230	2	104	62-150
12	22	10-62	9	75	26-220	4	138	90-270
13	11	6-20				5	106	82-135
			14	90	56-140	6	119	82-160
21	20	14-31	16	93	70-135	7	209	106-290
			17	46	36-82	8	161	64-280
						10	154	50-190
						11	152	120-220
						18	118	92-135
						19	187	135-290
						20	119	64-180

* Effluent chloride concentrations were 100 to 180 mg/l with a mean of 134 mg/l.

† Slight impacts, a mean of less than 32 mg/l Cl; moderate, a mean in the range 33-100 mg/l Cl; and great, a mean of more than 100 mg/l Cl.

adsorbed in the sand and gravelly soil horizons. Previous studies of soil characteristics at this site showed that chlorides were carried through the very permeable horizons in the percolating water.[20] Chloride levels in the native groundwater at the treatment site (wells 13 and 21) were 6 to 31 mg/l. Effluent chloride concentrations were 100 to 180 with a mean of 134.

Chloride levels in the groundwater samples indicated the relative influence of the percolate upon the groundwater quality and permitted each well to be placed in one of three arbitrary groups, according to the mean chloride levels (Table I). Native groundwater wells and the wells only slightly impacted by chloride had a mean chloride level less than 32 mg/l. Moderately impacted wells contained mean chloride levels of 33 to 100 mg/l and greatly impacted wells had mean chloride levels of more than 100 mg/l. Grouping the wells according to the relative influence of the percolate revealed that the wells with the highest mean chloride levels were located north and east of the treatment site, while moderately impacted wells were positioned along southern sides of the application area.

Although the groundwater gradient was about 1% towards well 12 from beneath the treatment site, chloride levels in well 12 indicated that the percolate movement in this direction was sufficient to increase the chlorides only occasionally. These data supported the groundwater gradient data and the conclusion that the groundwater flow at the treatment site was north and east from the site.

During the early spring of 1974, attempts were made to define the direction and rate of groundwater movement from the treatment site and possible dilution effects using the tritium tracer. Data shown in Table II strongly suggest that some tritium reached the wells north and northeast of the treatment basins which received the tritium-spiked effluent. However, because of the inconsistencies in the data, it was not possible to determine the time of travel or to predict the main flow lines. Highest tritium counts were detected in samples from wells 4, 5, 6, 7, 8, 10, 19 and 20, which were in the direction of expected groundwater flow based upon groundwater gradients and the chloride concentrations. About half of the samples taken from these wells suggest that spiked tritium had arrived but with considerable dilution, about 100 times or more. A dilution of this magnitude was inconsistent, however, with the chloride concentrations found in the effluent and samples from these wells. Comparison of the mean chloride levels of the effluent and the strongly impacted wells suggested a possible dilution of about 20% for some wells. In other wells, the chloride levels were greater than the effluent concentration, suggesting that no dilution had occurred.

Since the tritium-spiking operation had been carried out over a 4-day period, it seems unlikely that wells should have tritium above background levels one day and not the next. Although hydraulic conductivies

Table II. Tritium count rates above background for wells sampled on dates shown.

	Date (1974)							
Well	*28 Mar*	*29 Mar*	*30 Mar*	*31 Mar*	*1 Apr*	*2 Apr*	*3 Apr*	*6 Apr*
3	B	B	B	B	B	B	B	B
4	B	14	B	B	B	B	B	B
5	B	15	B	B	B	B	B	B
6	B	5	7	5	B	B	B	B
7	B	12	7	B	B	B	B	B
8	B	4	7	B	B	B	B	B
9	B	–	B	B	B	B	B	B
10	B	20	B	B	B	7	3	B
11	B	10	B	B	B	B	B	B
12	B	–	B	B	B	B	B	B
13	B	B	B	B	B	5	B	B
14	B	–	B	B	B	B	B	B
16	B	–	B	B	B	B	B	B
17	B	11	B	B	B	B	B	B
18	B	B	11	B	B	B	B	B
19	B	16	7	B	B			
20	B	21	12	6	6	12	B	B
21	B	B	B	B	B	B	B	B
N. pond	B	B	B	B	4	B	B	B

Note: B represents background levels measured using distilled water; the numbers represent actual counts above background levels. All counts were made for one hour.

Table III. Chemical and bacteriological characteristics of primary sewage effluent in 1974.

*Parameter**	*Mean*	*Range*	*Additions† (kg/ha day)*
pH (standard units)	6.9	6.0-7.7	–
BOD_5	107	42-172	147
COD	210	105-415	289
Total nitrogen	40	30-62	55
Organic nitrogen	19	13-29	26
NH_4-N	18	6-28	25
NO_3-N	3	0.7-7.0	4
Total PO_4-P	10	4-16	14
Ortho PO_4-P	7	3-13	10
Chlorides	134	100-180	184
Total coliform bacteria ($\times 10^6$/100 ml)	6.8	1.4-33	–

* mg/l unless otherwise indicated.

† Constituent addition based on mean concentration and mean flow of 4470 m^3/day from 4 January to 21 June 1974.

were determined in the laboratory by using permeameters for several layers in treatment basins 13, 14 and 15, whether these hydraulic conductivities were representative of the strata primarily responsible for carrying water away from the treatment site is not known. Conductivities measured in the laboratory suggest that the tritium tracer could have arrived at the peripheral observation wells within a few days, whereas less permeable horizons could require weeks to carry the tracer to the wells. From data shown in Table II, it seems reasonable that the tritium tracer had arrived at well 20 and at a few other wells, but not in sufficient quantity to predict tracer travel time or to predict main flow lines away from the treatment beds.

Although the tritium study was not conclusive, results did support other groundwater data in regard to the direction of groundwater flow and strongly indicated that wells north and northeast of the application site were correctly positioned to monitor groundwater movement from the treatment site.

The results of chemical and bacteriological analyses of the primary sewage effluent applied to the treatment basins from 4 January through 21 June 1974 are presented in Table III.

The quality of the effluent observed in the 1974 study was comparable to that observed during the 1973 studies. Effluent applied to the 9 treatment basins from 1 January to 21 June 1974 totaled about 27 m (89 ft). This volume of effluent was approximately the volume that had been applied during the previous 1 January through 31 December 1973 investigations: 27.1 m.

COD and BOD_5

COD and BOD_5 concentrations in the primary sewage effluent were 105 to 415 mg/l and from 42 to 172 mg/l, respectively. Under the 7-day effluent inundation, COD and BOD_5 additions totaled approximately 48,580 kg/ha and 24,750 kg/ha, respectively (43,390 and 22,100 lb/acre, respectively).

COD and BOD_5 concentrations in groundwater directly beneath the application area (well 2) were quite low, ranging from 10 to 50 mg/l COD and 0.5 to 9.4 mg/l BOD_5, with mean values of 27 mg/l COD and 5.7 mg/l BOD_5. Groundwater COD and BOD_5 levels in the observation wells located around the periphery of the treatment site were comparable to those observed in well 2. Mean COD levels were generally less than 30 mg/l, while mean BOD_5 levels were about 3 mg/l. These levels were about 14% and 4%, respectively, of mean COD and BOD_5 levels calculated for the primary effluent. The COD and BOD_5 concentrations in the peripheral wells were greater, however, than native groundwater levels (Table IV).

Nitrogen

Nitrogen in the primary effluent was about evenly divided between organic-nitrogen, 13 to 29 mg/l and NH_4-N, 6 to 28 mg/l, with a small amount of NO_3-N, 0.7 to 7.0 mg/l (Table III).

Total nitrogen levels in the effluent varied through the study period. During the July and August monitoring period, nitrogen levels increased slightly because of increased organic-nitrogen levels (Fig. 5). Effluent NO_3-N levels initially were 1 to 2 mg/l. Nitrate levels during April through mid-July were about 4 mg/l but decreased to 2 mg/l in September.

Total nitrogen inputs to the treatment basins from January through June were calculated at about 9250 kg/ha, 48% of which was organic-nitrogen; 45%, NH_4-N; and 7%, NO_3-N.

NO_3-N was the major nitrogen component in the groundwater, while organic nitrogen and NH_4-N were less than 2 mg/l. Exceptions were observed in wells positioned along the east edge of the application area. In these wells, the level of NH_4-N was about equal to the NO_3-N levels (Fig. 6).

Concentrations of total nitrogen in the native groundwater, in wells 1, 13 and 21, varied from 0.4 to 5.5 mg/l, with mean values of about 2 mg/l (Table V).

The total nitrogen levels in the observation wells illustrated the relative influence of the percolate. Wells slightly impacted with chloride had total nitrogen levels only closely comparable to native groundwater levels (1-5 mg/l), whereas moderately impacted wells had 3-10 mg/l nitrogen. The strongly impacted wells, positioned in the direction of groundwater movement, had the highest total nitrogen levels, averaging more than 10 mg/l throughout the study period (Table IV).

The total nitrogen and NO_3 levels observed in moderately impacted wells is represented by wells 3 and 14 (Fig. 7 and 8, respectively). During the latter part of the study period, nitrogen concentrations of these two wells increased for a short time, but then dropped off again. The nitrate peak early in July probably reflects the leaching of nitrates from treatment basins which had up to this time not received effluent during 1974. Additional NO_3-N peaks were not observed until the latter part of the study period, although treatment

Table IV. Groundwater chemical and bacteriological characteristics in selected observation wells, January-June 1974.

Parameter *	Well 2		Well 3		Well 4		Well 5		Well 6		Well 7		Well 8		Well 9	
	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range
pH (standard units)	6.6	6.4-7.7	6.4	5.7-7.0	6.8	6.5-7.5	6.7	6.4-7	7.6	6.6-8.1	5.9	5.1-7.1	6.9	5.7-7.1	6.4	6.0-6.9
BOD_5	5.7	0.5-9.4	1.3	0.2-4.1	2.3	0.5-6.9	3.3	0.4-7.7	1.4	0.1-4.4	2.3	0.2-8.5	2.3	0.02-8.1	1.7	0.1-5.5
COD	27	10-50	26	5-75	33	5-80	36	5-70	38	5-70	45	5-110	35	0.5-150	38	0.5-170
Total Nitrogen	18.3	11.6-26	9.4	1.8-32	20.8	10.5-41.6	16.8	5.9-27.4	21.1	10.3-24.6	18.2	3.6-32.9	20.7	8.4-40	6.9	1.6-39.4
Organic Nitrogen	7.1	0.2-10.4	1.0	0.2-2.4	7.9	4.4-14	6.2	2.4-13.6	8.0	2.8-10.6	1.7	0.2-5.2	1.8	0.2-3.6	[illegible]	0.2-4.4
NH_4-N	8.8	0.4-14	1.2	0.05-6	9.5	0.4-34	8.3	2.9-14.5	9.6	1.6-13	3.0	0.4-8.0	3.5	0.3-8.0	1.2	0.2-7
NO_3-N	2.4	0.9-5.3	7.2	0.9-30	3.4	0.8-9.5	2.3	0 6.6	3.5	0.7-11	13.5	0.8-32	15.4	1.5-38	4.2	1-29
Total PO_4-P	3.4	2.2-5.2	2.4	0.8-4.9	2.9	0.9-5.9	2.7	0.7-5.9	1.7	0.3-3.9	3.4	1.1-8.8	3.6	0.4-8.1	1.2	0.4-2.0
Ortho PO_4-P	1.4	0.4-3.3	0.5	0.05-1.5	0.4	0.01-1.2	0.5	0.01-1.3	0.3	0.01-0.8	0.4	0.02-1.6	0.5	0.02-1.8	0.1	0.01-0.6
Chlorides	104	62-150	92	42-230	138	90-270	106	82-135	119	82-160	209	106-290	161	64-280	74.6	26-220
Total Coliform Bacteria (#/100 ml)	3480	30-9700	21	0-220	5	0-20	10	0-80	85	0-100	539	0-8000	39	0-570	39	0-540

Parameter *	Well 10		Well 11		Well 12		Well 14		Well 16		Well 17		Well 18		Well 19		Well 20	
	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range	mean	range
pH (standard units)	5.9	5.5-6.9	6.2	5.9-7.4	6.5	6.0-7.8	6.4	6.2-6.8	6.0	5.8-6.3	6.3	5.9-7.5	6.2	5.9-7.4	6.0	5.7-7.2	6.0	5.7-7.2
BOD_5	2.0	0.2-5.3	1.1	0.2-4.4	0.9	0.02-4.3	1.3	.03-4.8	2.6	0.2-6.0	2.3	0.2-8.8	2.1	0.1-7.6	2.2	0.4-6.0	1.5	0.5-6.0
COD	27	5-65	32	2-125	23	4-75	22	0.5-75	26	1-100	8	0.2-25	20	1-45	16	0.5-65	26	0.5-160
Total Nitrogen	14.3	1.4-24.6	12.6	3-19.8	2.8	1.0-4.5	4.5	1.9-8.2	6.0	1.6-18.1	3.5	2.3-5.4	9.9	4-16.8	19.3	10.4-38.5	12.3	2.6-28
Organic Nitrogen	1.0	0.2-2.4	1.1	0.2-2.8	1.1	0.2-2.4	1.0	0.2-2.0	3.7	0.2-16	1.0	0.2-2.0	1.2	0.2-2.4	2.2	0.2-6.8	0.8	0.2-2.0
NH_4-N	0.3	0.1-1.0	0.7	0.1-3.6	0.2	0.01-0.6	0.3	0.02-0.6	0.5	.1-1.2	0.4	0.01-2.0	1.1	0.5-2.9	0.2	0.02-0.5	0.3	0.02-1.1
NO_3-N	13	5.6-24	10.8	4-17	1.5	0.4-2.6	3.2	1.4-6.2	1.8	0.4-5	2.1	1.4-3.2	7.6	2.1-15	16.9	8.4-36	11.2	2.1-25
Total PO_4-P	1.8	0.4-3.6	2.4	0.3-8.2	1.0	0.1-2.0	1.7	0.3-4.9	2.2	0.7-6.2	1.8	0.3-7.8	2.1	0.5-4.9	3.0	0.5-7.8	1.5	0.4-3.9
Ortho PO_4-P	0.08	0.01-0.6	0.2	0.01-0.8	0.07	0.01-0.4	0.1	0.01-0.5	0.4	0.01-1.2	0.2	0.01-0.8	0.2	0.01-1.1	0.5	0.01-2.8	0.1	0.01-0.5
Chlorides	154	50-190	152	120-220	22	10-62	90	56-140	93	70-135	46	36-82	118	92-135	187	135-290	119	64-180
Total Coliform Bacteria (#/100 ml)	16	0-100	24	0-150	10	0-70	15	0-100	44	0-350	65	0-705	13	0-60	3	0-20	13	0-140

* mg/l unless otherwise indicated

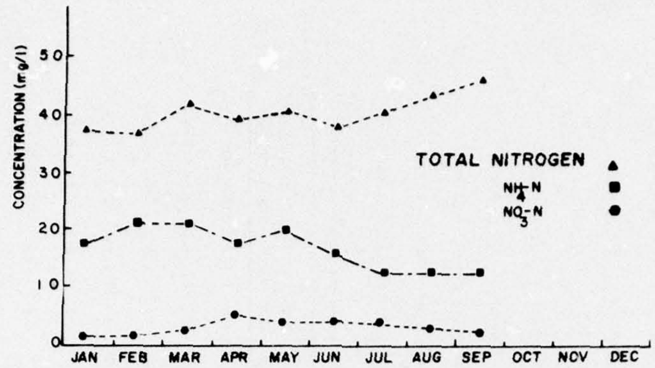

Figure 5. Total nitrogen, ammonium nitrogen and nitrate nitrogen in primary sewage effluent in 1974.

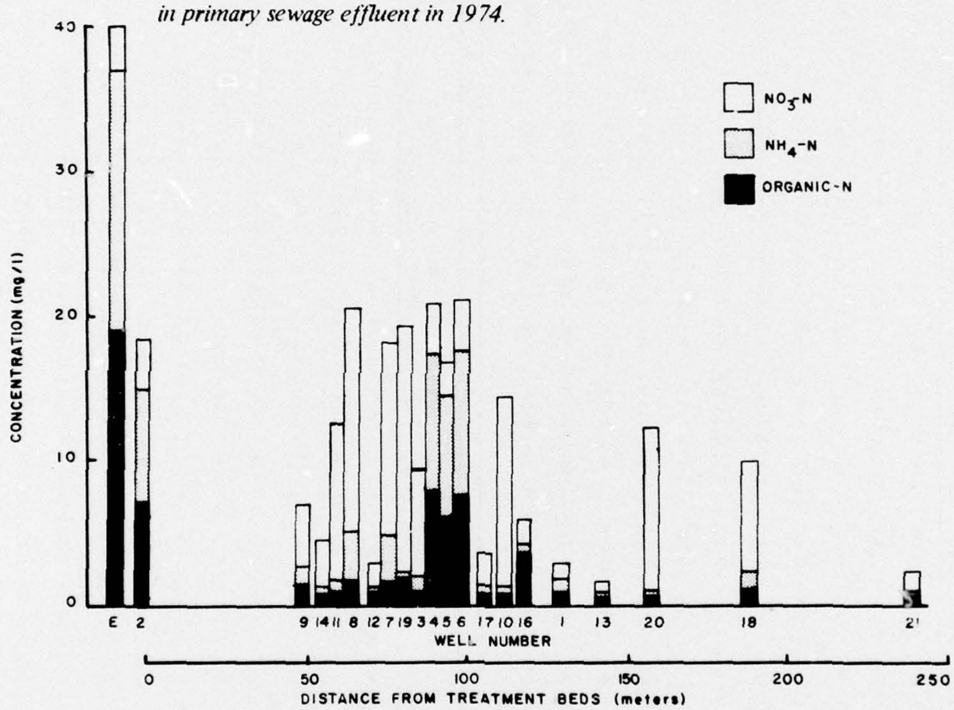

Figure 6. Organic nitrogen, ammonium nitrogen and nitrate nitrogen in primary effluent and 20 wells, January-June 1974.

basins were being inundated and rested in rotation. The second nitrate peak corresponded with the renewed effluent application to the original 9 treatment basins, which had been rested for 60 consecutive days.

The nitrogen concentrations observed in wells 8 and 10 depicted the nitrogen levels in the strongly impacted wells, which had mean total nitrogen levels of more than 10 mg/l (Fig. 9 and 10, respectively). Additions of nitrogen to the treatment basins during the 1973 studies totaled about 12,900 kg/ha yr, which was approximately one-third less than the 1974 rate, 20,100 kg/ha yr. The primary reason for the greater nitrogen additions in the 1974 study was the longer inundation period, which resulted in a greater total volume of effluent being applied during each 7-day application period, since the effluent application rates, length of the recovery period, and effluent qualities were closely comparable for both the 1973 and 1974 studies.

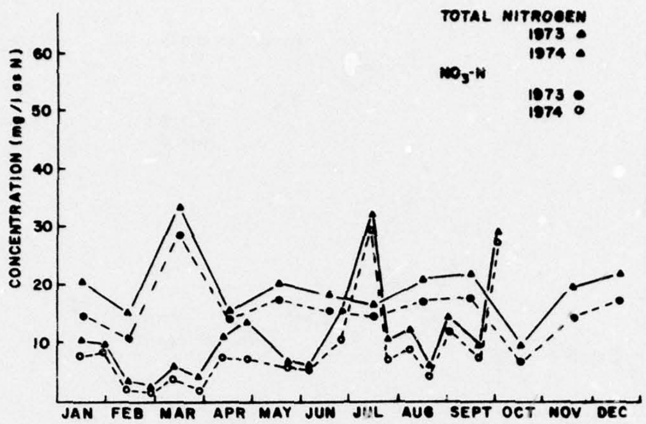

Figure 7. Total nitrogen and nitrate nitrogen in well 3, 1973-1974.

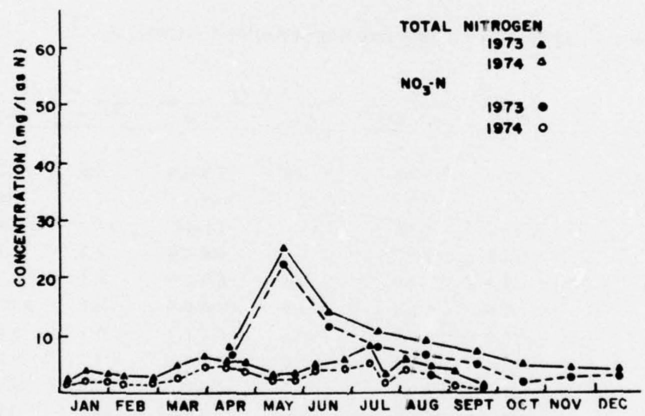

Figure 8. Total nitrogen and nitrate nitrogen in well 14, 1973-1974.

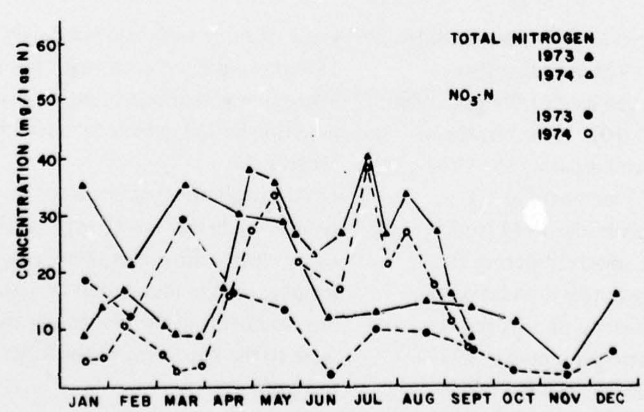

Figure 9. Total nitrogen and nitrate nitrogen in well 8, 1973-1974.

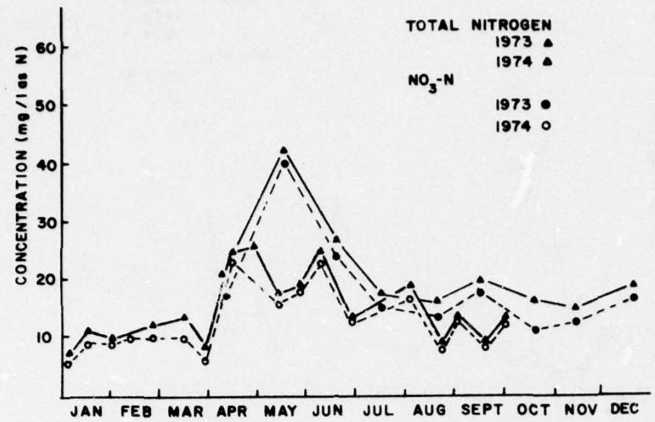

Figure 10. Total nitrogen and nitrate nitrogen in well 10, 1973-1974.

Table V. Qualitative characteristics of native groundwater.

	Well 1		*Well 13*		*Well 21*	
*Parameter**	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
pH (standard units)	7.5	7.1-7.8	6.0	5.5-6.8	6.6	6.2-7.3
BOD_5	4	2-5	1	0.04-5	2	0.4-6.0
COD	40	19-90	11	1.0-50	12	0.5-45
Total nitrogen	2.5	1.0-5.5	1.8	0.4-4.4	2.3	0.7-5.1
Organic nitrogen	0.8	0.2-2.0	0.8	0.2-2.0	0.9	0.2-2.0
NH_4-N	0.6	0.1-1.9	0.3	0.02-0.6	0.2	0.02-0.5
NO_3-N	1.0	0.4-3.2	0.6	0.1-2.7	1.3	0.4-4.7
Total PO_4-P	3.4	1.4-6.2	1.0	0.3-1.6	1.7	0.2-4.2
Ortho PO_4-P	1.2	0.2-3.6	0.1	0.01-0.4	0.05	0.01-0.3
Chlorides	31	24-52	11	6-20	20	14-31
Total coliform bacteria (no./100 ml)	290	0-3000	10	0-70	2	0-10

* mg/l unless otherwise indicated.

Comparison of total nitrogen levels in the groundwater for both the 1973 and 1974 investigations revealed that groundwater nitrogen levels were generally less for the 1974 study (Fig. 7-10). However, the nitrogen levels were greater in well 8 during the 1974 investigations than in the 1973 studies (Fig. 9).

Groundwater nitrogen levels in the 1974 studies were generally less than those observed during the comparable 1973 period. The longer inundation period may have increased the quantity of nitrogen removed in the 1974 study. Nitrogen inputs in 1974 were 54% greater than those in 1973, but a correspondingly greatly increase in groundwater nitrogen levels was not observed. The nitrogen in the groundwater of some wells increased only 1 to 4 mg/l or 6 to 15% above the previous year's levels. The net loss of nitrogen was assumed to be the result of the longer inundation period, which enhanced the denitrification process.

Although nitrogen levels for some wells were not reduced to below the 10 mg/l total N, results of this study clearly show that more nitrogen was removed by lengthening the inundation period. However, these improvements in the percentage of nitrogen removed were to the detriment of the infiltration capacity of the basin.

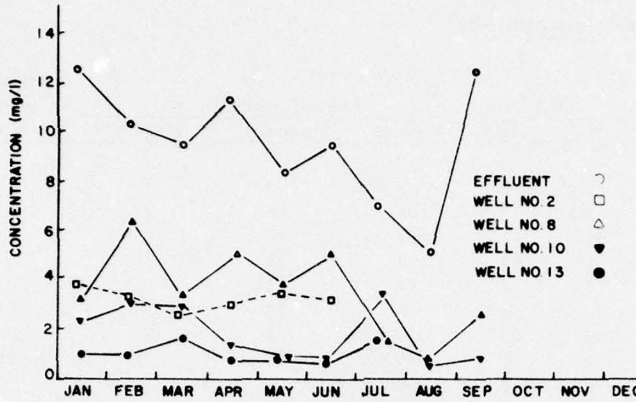

Figure 11. Total phosphorus in primary sewage effluent and observation wells 2, 8, 10 and 13.

Phosphorus

Total phosphorus in the primary sewage effluent was 4 to 16 mg/l PO_4-P, with a mean value of 10 mg/l PO_4-P. Readily available phosphorus as orthophosphate was 3 to 13 mg/l PO_4-P, with a mean of 7 mg/l. Ortho PO_4-P constituted about 70% of the total 2310 kg/ha PO_4-P applied to the treatment basins during this study.

Total phosphorus levels in the effluent varied throughout the study, but in general, they declined over the study period (Fig. 11).

Although substantial quantities of phosphorus were applied to the 9 treatment basins, phosphorus concentrations in the groundwater directly beneath the application site (well 2) were 2.2 to 5.2 mg/l total PO_4-P, which was about 34% of the effluent PO_4-P concentration. Ortho PO_4-P was 0.4 to 3.3 mg/l, with a mean of 1.4 mg/l. The orthophosphorus concentrations were about 20% of the ortho PO_4-P concentration in the primary sewage effluent.

Total PO_4-P concentrations in the wells strongly impacted by the percolate, and positioned north and east of the application site, were closely comparable to the concentrations beneath the treatment sites. The mean PO_4-P levels in these wells were 2.0 to 3.6 mg/l, and the ortho PO_4-P concentrations varied from 0.1 to 0.5 mg/l. The monthly mean PO_4-P levels for the strongly impacted wells (2, 8 and 10) showed that groundwater PO_4-P levels were 10-35% of the effluent concentration.

Phosphorus additions to the basins during 1974 were substantially greater than PO_4-P inputs during 1973, 55 kg/ha day and 36 kg/ha day, respectively. Even so, groundwater phosphorus levels for the two years showed that the 1974 total phosphorus levels were only 1 to 2 mg/l higher when basins were inundated for longer periods.

Coliform bacteria

Total coliform bacteria in the primary sewage effluent applied to the treatment beds was 1.4×10^6 to 33×10^6/100 ml and averaged 6.8×10^6/100 ml. Although coliform counts in the primary effluent were substantial, coliform bacteria were essentially removed from the effluent upon percolation through the 18.3 m of stratified sand and gravel. Coliform counts directly beneath the application area were 30 to 9700/100 ml and averaged approximately 3500/100 ml. Although the total coliform counts were rather high in well 2, analysis for fecal coliform bacteria proved negative. It was suspected that the counts of total coliform bacteria were indicative of indigenous soil bacteria.

Coliform bacteria counts in the samples from the other wells were less than 100/100 ml, which was only slightly larger than background levels. In terms of coliform bacteria, the groundwater quality in these wells met "raw" potable water quality standards.

Overall comparison of the treatment efficiency of the Fort Devens system as compared with conventional treatment processes is presented in Table VI. As

Table VI. Water quality characteristics of effluents from primary, secondary and tertiary wastewater treatment, potable drinking water, and groundwater for selected wells at the Fort Devens wastewater treatment site.

Parameter*	Well		Level of treatment			Drinking water
	8	10	Primary	Secondary†	Tertiary**	criteria††
pH (standard units)	6.9	5.9	6.9	7	7	5-9
BOD_5	2.3	2.0	106	25	5	
COD	35	27	210	70		
Total nitrogen	21	14	40	20	32	
Organic nitrogen	2	1	19	2		
NH_4-N	4	0.3	18	10	2	0.5
NO_3-N	15	13	3	8	30	10
Total PO_4-P	4	2	10	10	0.5	No limit
Ortho PO_4-P	0.5	0.1	7		0.5	
Chlorides	161	154	134	100		250
Total coliform bacteria (no./100 ml)	39	16	7×10^6		200	1×10^4

* mg/l unless otherwise indicated.
† Reed et al.[18]
** Extended aeration of activated sludge with alum polyelectrolyte coagulation, sedimentation, and filtration.
†† Environmental Protection Agency.[8]

indicated, treatment efficiency was better in general than secondary treatment efficiency and was comparable in most instances to that of conventional tertiary treatment.

CONCLUSIONS

1. In the treatment of wastewater by rapid infiltration, the operational cycle of an inundation period followed by a recovery time is critical for the optimization of natural nitrogen removal mechanisms. In this study, unchlorinated primary effluent was applied at the rates of 0.42 to 0.57 m/day with an operational cycle of 7 days of inundation followed by 14 days of recovery.

2. The groundwater quality surrounding the treatment site showed that 5-day biochemical oxygen demand, chemical oxygen demand, and total coliform bacteria were essentially removed from the unchlorinated primary sewage effluent upon infiltration and percolation of the effluent through 18.3 m of sandy soil.

3. When the treatment basins were inundated for 7 days, the percentage of total nitrogen removed from the unchlorinated primary effluent was greater than when the basins were inundated for 2 days. However, after 6 months, the infiltration capacity had been reduced so that the basin surfaces were still wet at the beginning of the next cycle of inundation and recovery.

4. An extended recovery period of 60 consecutive days rejuvenated the infiltration capacity of the treatment basins so that the 7-day application/14-day recovery cycle could once again be used. This restoration of infiltration capacity is attributed to oxidation of accumulated organics during the extended recovery period.

5. The design of rapid infiltration systems to take advantage of greater nitrogen removal through longer inundation and recovery cycles can be accomplished by allowing an extended recovery period for the oxidation of accumulated organics. Determination of the length of inundation/recovery cycle, number of cycles, and length of recovery period is directly related to the requirements of the specific site.

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